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Monitoring and Assessment of the Marine Benthos at UK Dredged Material Disposal Sites

Prepared by the Benthos Task Team for the Marine Pollution Monitoring Management Group Coordinating Sea Disposal Monitoring

Scottish Fisheries
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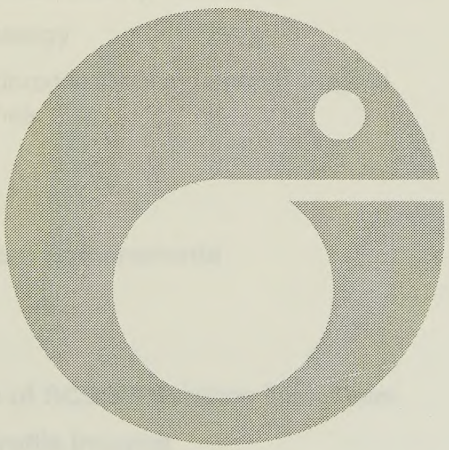

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Monitoring and Assessment of the Marine Benthos at UK Dredged Material Disposal Sites

(Prepared by the Benthos Task Team for the MPMMG Group Coordinating Sea Disposal Monitoring (GCSDM). Membership of this Task Team is given in Appendix 1)

SOAEFD Marine Laboratory
Aberdeen

Introduction

This report reviews current knowledge of the biological impacts on and in the sea bed arising from dredged material disposal, develops criteria for judging the acceptability of any impacts according to agreed Environmental Quality Objectives and Standards, and makes recommendations concerning the routine monitoring and assessment of dredgings disposal sites.

There are three key differences between the disposal of dredged material and that of sewage-sludge (Rees et al., 1990; MAFF, 1993a), which affect both field sampling strategies and the interpretation of results from field surveys:

1. The nature of the dumped material may range from rock to silts and clays and will contain a much higher solids content than sewage sludge, which rarely exceeds 6%,

2. The frequency of disposal at a licensed site may vary over a much wider time-scale (from days to years) and

3. The literature base dealing with the biological consequences of dredgings disposal is more limited than that for sewage-sludge disposal.

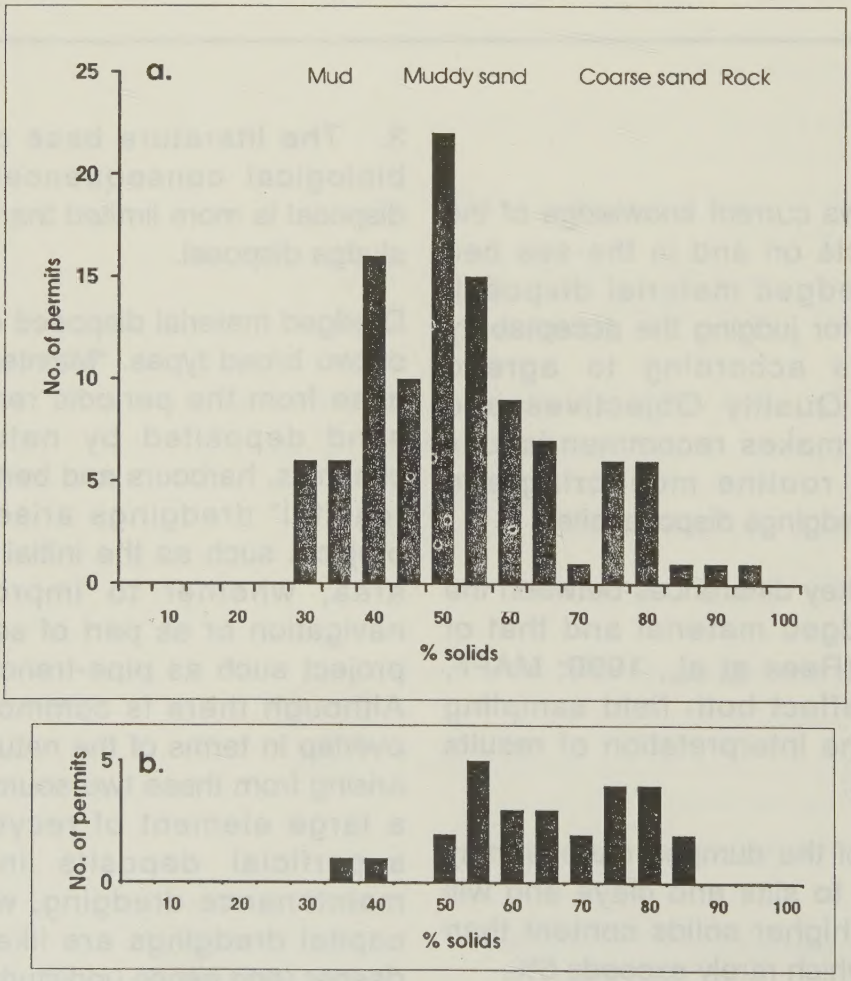
Dredged material disposed of at sea consists of two broad types. "Maintenance" dredgings arise from the periodic removal of silt and sand deposited by nature in dredged channels, harbours and berthing areas, while "capital" dredgings arise from "one-off" projects such as the initial deepening of an area, whether to improve depths for navigation or as part of some construction project such as pipe-trench or breakwater. Although there is commonly a degree of overlap in terms of the nature of the material arising from these two sources, there may be a large element of recycling of shifting superficial deposits in the case of maintenance dredging, while the bulk of capital dredgings are likely to arise from deeper (and hence undisturbed) deposits.

The dredging industry (contractors/port authorities/consultants) works to guidelines laid down by the Permanent International

Association of Navigation Congresses (PIANC). Included in these guidelines is a definition of dredged materials, based on particle size, which identifies six categories viz clay, silt, sand, gravel, cobbles and boulders. These categories will be applied in this work wherever possible but it must be recognised that a much wider range of definitions is accepted in the biological sciences.

As with sewage sludge, the disposal of dredged material at sea in United Kingdom waters is licenced under the Food and Environment Protection Act 1985 Part II (FEPA), following guidelines laid down by International Conventions (Great Britain — Parliament, 1972a, b and 1985). Criteria which must be satisfied before a licence is issued include the chemical quality of the material, the quantity to be disposed of and its nature and origin. (Sediments containing unacceptably high levels of contaminants will not be licensed for sea disposal). Presently, there is no requirement for particle size analyses of samples to be conducted, but a general indication of variability in the type of dredged material licensed for sea disposal around the United Kingdom coastline may be inferred from the % solids content derived from licence applications in 1991 (Fig. 1a and b). The bulk of maintenance dredgings consisted of silty sands, while capital dredgings were proportionately coarser in nature and, on occasion, included significant quantities of large cobbles and boulders. The total amounts of maintenance and capital dredgings disposed of to sea in 1991 were, respectively, 29 and 13 million wet tonnes.

A general review of the types of dredging activity, sources and nature of the material to



Figures 1a-b. Frequency distribution of total solids in maintenance (a) and capital (b) dredgings licensed in 1991. (No particle size data are available, and hence descriptions of substrate type are only broadly indicative).

be disposed of, and other relevant studies, was followed by attempts to develop descriptive "models" describing changes in the benthos in response to different types of dredged material, as far as current knowledge permits. This was considered to be a pre-requisite for assessment of the biological consequences of solid waste disposal in relation to the type (and magnitude) of the disposal operation. For example, while all such activities demonstrably induce a change to the environment at the disposal ground, it is by no means the case that all such activities induce adverse effects in the longer term, even in cases where substrates are materially altered as a result of incorporation of deposited material.

Recommendations are made concerning future monitoring strategy, and requirements for the application of "Environmental Quality Standards" at UK dredgings disposal sites. Finally, a list of future needs, including Research and Development requirements, is presented.

This report deals solely with effects associated with the **disposal** of dredged material at designated sites, and not with those at the point of dredging. However, a number of parallels may be drawn between the two activities in terms of the biological consequences of man-made physical disturbances. (See Campbell, 1993 for a summary of requirements for the scientific evaluation of the consequences of marine aggregate extraction).

General Review of Dredging and Disposal Practices

Nature of Dredged Material and Dredging/Disposal Methods

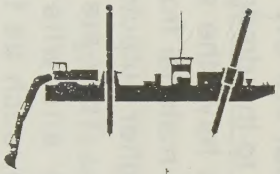
There is much greater variability in the physical and chemical nature of material arising from dredgings disposal — and hence the ecological impacts — than with sewage sludge. The material to be removed in a dredging operation may range from solid rock, dislodged by blasting, through consolidated boulder clay and cohesive recent clays to sands and even fluid silt. Whatever the original nature of the material being dredged, the physical form of it at the time of discharge varies greatly according to the type of dredging and transport plant employed.

There are six basic types of dredging plant (Fig. 2). Grab dredgers come in two forms, pontoon-mounted grab cranes and self-propelled grab hopper dredgers. Large pontoon-mounted grab cranes are used by dredging contractors as the large capacity grabs have the ability to excavate a wide range of material, including broken rock and debris such as old piles. Self-propelled grab hopper dredgers are owned by both port authorities and dredging contractors and have hopper capacities ranging from about 100 to over 1,500 cubic metres. They are mainly used for maintenance dredging, especially of small harbours, alongside quay walls and in other areas where trailer hopper dredgers cannot easily work.

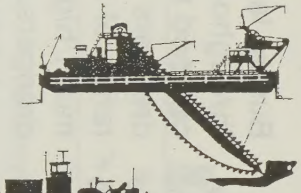
Grab dredgers have the ability to dredge material and place it in a hopper at near enough the in situ density, and it is quite possible that relatively large cohesive lumps of material may be released at the disposal site.

Backhoe dredgers consist of hydraulic excavators mounted on a pontoon which is equipped with heavy vertical legs or spuds to keep the pontoon in position and provide resistance to the horizontal digging forces. Backhoe dredgers are owned by dredging contractors and used where stiff material such as boulder clay or soft rock is to be removed. Material can be dredged at its in situ density and is placed in a hopper barge for disposal.

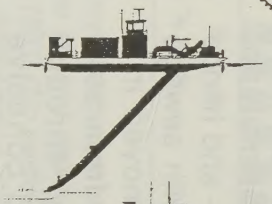
Bucket ladder dredgers were once common throughout the UK but examples are now confined to the dredging contractors. Bucket dredgers have the ability to dredge a wide range of material from broken rock to silt at much higher outputs than grab or backhoe dredgers. They are used where material is variable and where some precision is required, such as the dredging of a pipe-trench. While material can be dredged at



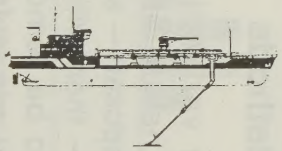
Backhoe dredger: A stationary dredger, moved on anchors or on spudpoles. A spud is a large pole that can anchor a ship while allowing a rotating movement around the point of anchorage. Small backhoe dredgers can be track mounted and work from the banks of ditches. A backhoe dredger is a hydraulic excavator equipped with a half open shell. This shell is filled moving towards the machine. Usually the dredged material is loaded in barges. This machine is mainly used in harbours and other shallow waters.



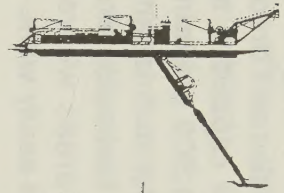
Bucket dredger: A stationary dredger, fixed on anchors and moved while dredging along semi-arcs by winches. The bucket dredger is one of the oldest types of dredging equipment. It has an endless chain of buckets, that fill while scraping over the bottom. The buckets are turned upside down and empty moving over the tumbler at the top. The dredged material is loaded in barges.



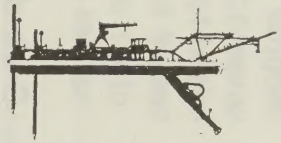
Water injection dredger: A self propelled dredger which brings the sediment to be excavated into suspension with waterjets. This suspension is denser than water. It will be carried away by gravity and currents. Water injection dredging is mainly used for maintenance in harbours.



Trailing suction hopper dredger: A self propelled ship which fills its hold or hopper during dredging, while following a pre-set track. The hopper can be emptied by opening bottom doors or valves (dumping) or by pumping its load ashore. This kind of dredger is mainly used in open water: rivers, canals, estuaries and the open sea.

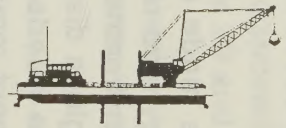


Suction dredger: A stationary dredger used to mine for sand. The suction pipe is pushed vertically into a sand deposit. If necessary water jets help to bring the sand up. It is loaded into barges or pumped via pipeline directly to the reclamation area.



Cutter suction dredger: A stationary dredger which makes use of a cutter head to loosen the material to be dredged. It pumps the dredged material via a pipeline ashore or into barges. While dredging the cutter head describes arcs and is swung around the spudpole powered by winches. The cutter head can be replaced by several kinds of suction heads for special purposes, such as environmental dredging.

Reclamation dredger: A stationary dredger used to empty hopper barges. A suction pipe is lowered into the barge. Extra water can be added by water jets to facilitate the suction process. The dredged material is pumped by pipeline ashore, to a reclamation area, or to a storage depot.



Grab dredger: A stationary dredger, moored on anchors or on spudpoles. The dredging tool is a grab normally consisting of two halfshells operated by wires or (electro)-hydraulically. The grab can be mounted on a dragline or on a hydraulic excavator of the backhoe type. Many modifications of grabs have been constructed like (top)open grabs, (top) closed grabs and watertight grabs. The grab dredger is used in harbours: the dragline type also in deep water. The dredged material is loaded in barges.

Figure 2. Types of dredger (from Donze, ed., 1990).

close to its in situ state, considerable mixing and break-up may take place in the transfer to the hopper barge.

Cutter suction dredgers remove material by suction action, assisted by mechanical loosening at the suction mouth. They are mainly employed by dredging contractors for capital dredging, especially in association with land reclamation. They are also used for trench dredging where the material may be placed in a temporary stockpile for later use as backfill. Material is discharged through a pipeline. Very occasionally cutter suction dredgers may load barges. They may also be used as rock fragmenters, breaking up rock for later dredging by other types of equipment.

Most dredging today is undertaken by trailer suction hopper dredgers. These are owned by both port authorities and dredging contractors and have hopper capacities from a few hundred to over 15,000 cubic metres. They operate like floating vacuum cleaners and are best suited to dredging silt and sand from navigation channels, although they may also work within docks and basins. The suction action may be assisted by water jets or teeth and generally the dredged material in the hopper is very well mixed and, in the case of silts, usually diluted. The hopper is emptied either through bottom doors or valves, or by pumping. The latter method is employed where the material is placed ashore or used for beach nourishment.

Some hopper dredgers and most hopper barges are of the split type, where material is released by the hopper splitting into two halves. This gives a much faster release than through bottom doors or valves, especially with sand and sticky clay.

Trailer dredgers may be assisted by underwater levellers or scrapers which move materials from areas such as close to quay walls and between mooring dolphins to where the trailer can dredge it. These devices are operated from small tugs.

Water injection dredgers inject water into silt deposits and by fluidising the silt cause it to move as a density current into deeper areas or into tidal streams. The material is well dispersed by this technique.

Sea Disposal

Virtually all offshore sea disposal takes place by "instantaneous" discharge from the underside of the transport vessel. In contrast to sea disposal of sewage sludge which is dispersed into the wake of a moving vessel, dredged material tends to sink rapidly as a single mass made up of the bulk of a cargo (see Fig. 3). Also by way of contrast, large quantities of material (perhaps one million tonnes in a capital dredging project) may be disposed of in a very short space of time (weeks or months), and the nature of the material may be very variable, ranging from "soupy" silt through sand to "stiff" clay and rock.

The settled mass will then disperse over time, depending on its cohesive strength. A disposal ground may frequently be characterised by the occurrence of discrete elongate mounds that represent individual barge loads (Fig. 4). These can retain their identity for many months in the case of boulder clay, and probably almost in perpetuity in the case of rock.

Disposal by pipeline is most often practised in harbours and estuaries when suction dredgers are employed. Disposal may be to landfill, to intertidal areas or by side-casting to the side of a channel. Beach nourishment projects often use pipelines to emplace offshore dredged material, as do sub-sea trenching projects prior to pipe-laying.

Quantity and Intensity of Disposal

Maintenance dredgings

Continuous disposal of maintenance dredgings by a port authority's own full time plant used to be the normal practice even at relatively small ports. This resulted in a steady discharge rate over time, with significantly larger amounts for sea disposal only arising periodically as a result of capital dredging projects. In many places, this practise has been replaced by intermittent

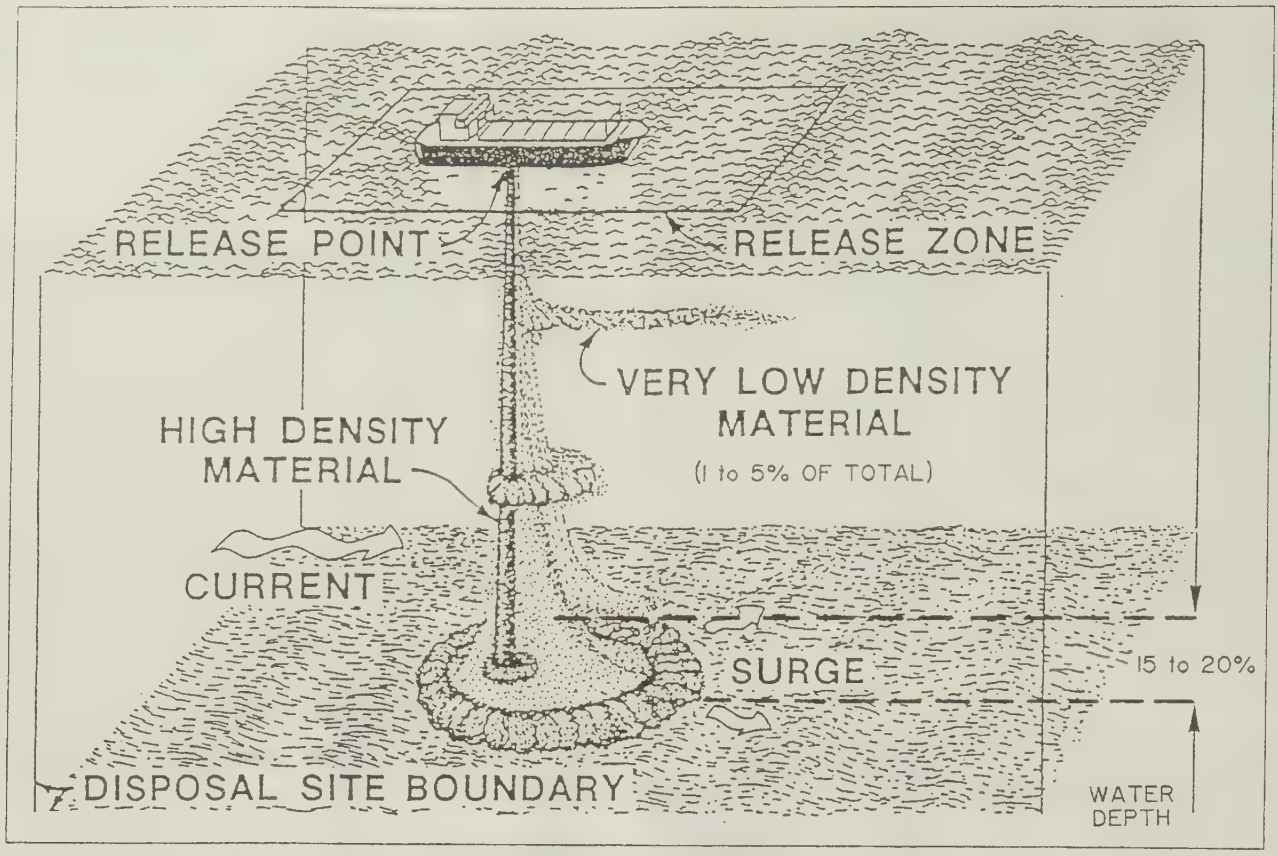


Figure 3. Transport processes during open water disposal (from Collins, 1990 after Truitt, 1988 and earlier authors).

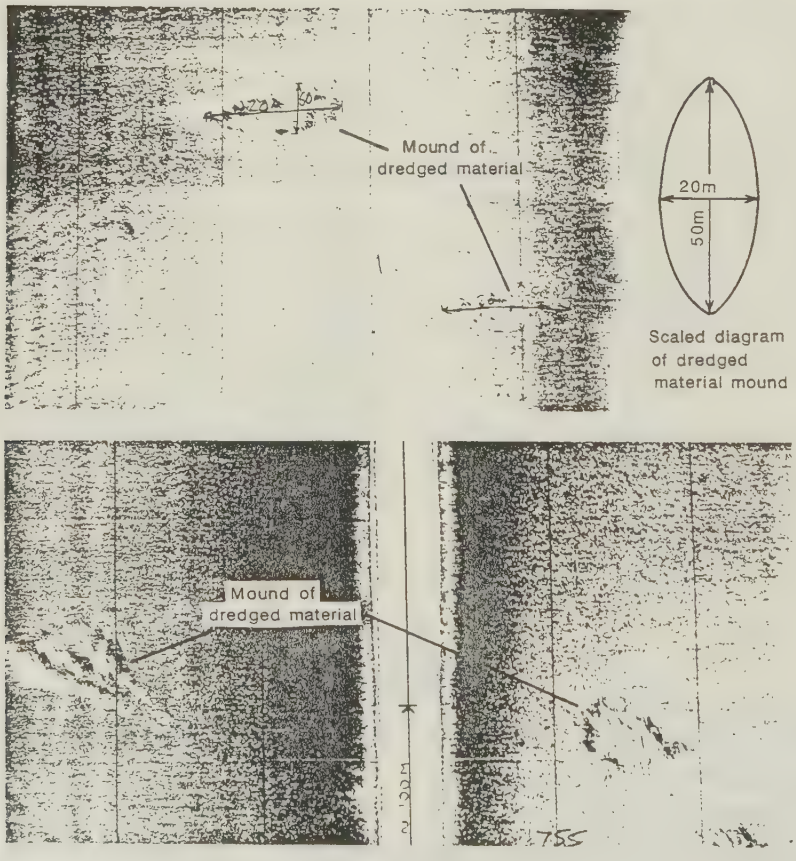


Figure 4. Sidescan sonar images of mounds of dredged material on the sea bed at a disposal site in the outer Thames estuary (from Rowlatt and Limpenny, 1987).

dredging by specialist contractors. Although the total quantity disposed of may often be less now than previously, the ecological effects may be greater, since the total amount previously going to sea in, say, 3-5 years, may now go to sea in a matter of only a few weeks.

Capital dredgings

Technological advances in the dredging industry, as well as changes in transportation patterns, increasingly establish dredging as the best practicable environmental option for large projects. These include not only familiar activities such as port developments, but also pipeline landfalls and estuary crossings. A notable example of the last activity was the construction of a submerged-tube road tunnel by means of trenching across the estuary of the River Conwy, Wales.

Disposal Sites

Disposal sites for many of Britain's major estuarine ports were traditionally placed as close as possible to the port facilities, often irrespective of the fact that net bedload transport could be towards the site of dredging. Capital dredging operations, involving a higher proportion of coarse material, used sites which were generally located further offshore. Historically, the main considerations for both activities were operational rather than environmental, although there was a tendency to avoid the disposal of material that could provide obstructions on favoured trawling grounds, and where there might be implications for navigation.

"Near-field" and "Far-field" Effects

In general, only a small proportion of the total quantity of dredgings disposed of to sea will be biodegradable (even in the case of arisings from organically-enriched estuarine locations) and with time, the very high particulate load will ultimately be incorporated into bottom deposits. In conceptual terms, dredged material disposal involves "sources", "sinks" and transport paths between the two. Ultimate "sinks" may be at a considerable distance from the "sources", with complex patterns of deposition and resuspension during advection.

Naturally turbid estuaries have ecosystems that are pre-adapted to cope with ephemeral deposition and resuspension of dispersing fines, notably as a result of tidal erosion/deposition cycles. Naturally clear water ecosystems are not, and so small amounts of additional fines may have significant effects, eg on coral and kelp forest communities.

In the British context, most ports that require frequent maintenance dredging are in turbid estuaries. The disposal sites, advection paths and sinks are often within the ambit of the same estuaries. There are, however, a number of mainly west coast situations where the ports are bounded by clear-water rocky coasts. Dredgings disposal often takes place outside the port inlets and is often into areas with strong tidal currents. The effect here is to insert quantities of fines into regimes where they may easily be dispersed, but in the process of dispersion they may get carried relatively large distances to impinge on potentially sensitive communities. Even if only a small proportion of the total quantity of mobilised fines settles on a clean gravel community, on a rich sponge/bryozoan epifaunal community on boulders or on the fronds of sub-littoral algae, significant ecological effects might occur. Should such "far field" effects arise in areas of fisheries or conservation importance they may be deemed unacceptable as well as breaching standards for "no effect" outside immediate dispersal zones. (It should be emphasised that while concern for such an eventuality is felt to be legitimate, the only evidence to support a link between the settlement of fines in an area of conservation interest and dredgings disposal is anecdotal).

Modelling techniques may allow the initial dispersal patterns of the bulk of dredged material to be forecast with reasonable accuracy; however, erratic "far field" settlements (especially to ecologically "sensitive" locations) are likely to be much more difficult to predict. It is therefore

concluded that special consideration should be given to improvements in methodology for predicting the long-term fate of fine particulates, as well as to mineralogical or other techniques for identifying the origin of material in cases where there is a possibility of untoward settlement of fines in "sensitive" locations distant from dredgings disposal grounds.

Responses of Benthic Communities to Different Waste Types

Biological Impacts Arising from the Marine Disposal of Muddy Dredgings

Published information relevant to the assessment of the impact of the disposal of muddy dredgings can be divided into three areas, namely "inorganic", "organically-enriched" and "contaminated" wastes.

In general, impacts arising from the disposal of wastes are likely to be most clearly evident in "accumulative" areas, ie those characterised by relatively weak tidal currents and limited exposure to wave action. Discharge to dispersive areas does not generally appear to cause major, long-term, disruption (eg Van Dolah, 1984).

Inorganic waste

Three studies are notable for providing information on effects on the benthos of the disposal of fine inorganic waste to UK coastal waters, although none deals directly with dredgings disposal.

Probert (1975), along with earlier workers, studied the effects of china clay waste on the benthos of the predominantly gravel substrates of Mevagissey Bay, SW England. The discharge of waste via the White River led to a marked change in the sediment composition of the bay, such that there was a central area with >80% silt/clay, surrounded by an area which gradually declined in silt/clay content back to a base level of <20%. These changes in sediment composition had a marked effect on the benthos. An afaunal area developed and the fauna of the whole central area of the bay was less diverse than that found in the surrounding gravels and muddy sands.

The work of Bamber (1984) on the benthos of a fly ash disposal site off the north east coast of England is relevant to the study of the impact of the discharge of inert material. Disposal rates were such that at the centre of the site, up to three metres thickness of waste were deposited in each year. As might be expected, this led to an obvious impact, analogous to that seen in Mevagissey Bay. All the measured and calculated faunistic parameters correlated negatively with ash content.

Four zones were defined:

i) a zone of maximum pollution virtually devoid of macrobenthic life;

ii) a polluted zone with a severely depressed fauna;

iii) a subnormal zone;

iv) a clear zone.

There was no evidence of an enhancement in benthic populations on the fringes of the impacted area. Neither was there evidence of the occurrence of "pollution indicator" species in response to fly ash disposal. However, there was evidence of a "classical" response (ie a decline in diversity and an increase in abundance) around a dredgings disposal site associated with the fly ash disposal area.

The conclusions from an annual study of the impact of a solid waste discharge arising from the mining of potash off the NE coast of England (Craig and Shillabeer, 1993) were:

i) in the immediate vicinity of the discharge, the benthic fauna declined to very low levels (or even absence, on two sampling occasions). This decline occurred gradually over a period of twenty years, but was most apparent just after step changes in production levels;

ii) at the stations where the impact was most marked, there was no initial "enhancement" phase, ie the fauna began declining immediately in response to a build-up of waste material;

iii) there was some change in community composition at stations that were not grossly impacted, but at which sediment particle size distributions had changed. Species typical of siltier sediments were more apparent. These changes were not reflected in univariate measures such as total abundances (A) and total numbers of taxa (T).

A summary of changes in A, T and A/T ratio is given in Figure 5a-c for three selected stations sampled annually in March over the period 1970—90. Station 33 is closest to the discharge and, apart from a slight reversal in the mid-1980s, has shown a consistent trend of decline. There does not appear to have been any enhancement phase at this station. No consistent trends are evident at stations 56 and 83, further away from the discharge. Classification analyses of the 1985—89 data also put these stations outside the impacted area.

Organically-enriched dredgings

Intuitively, it might be expected that the response of benthic communities to the disposal of organically-enriched dredgings would be similar to the response to sewage-sludge or other organic waste disposal (see, eg, Pearson and Rosenberg, 1978; MAFF, 1993a). There is evidence in the published literature that this is indeed the case. Oliver et al. (1977) described the impact of dredgings disposal in Monterey Bay, California. They reported that most of the fauna was destroyed at the centre of the disturbance. However, with time, successional changes in the composition of the fauna occurred in the direction of a stable, pre-disturbance community.

Contaminated dredgings

Extensive studies of the impact of the disposal of highly contaminated dredgings off the mouth of the Duwamish river near Seattle, USA have included benthic surveys and studies on other bio-indicators (Geyer, 1981; Long, 1982; Stein et al., 1992). In this area, highly contaminated dredged material containing PCB's, PAH's and metals was dumped into 60 m of water creating a disposal mound which gradually spread during the post-disposal period. There was no significant uptake of PCB's or metals by animals inhabiting the disposal area, or by caged animals held in situ for three weeks. Changes were noted in the abundance, diversity and species composition of the benthos in the disposal area; however, similar changes in the reference area populations make it unlikely that disposal was wholly responsible for the changes.

Two important conclusions arise from this limited review:

i) the response of the benthos to inert fine particulates is not accompanied by an initial "enhancement" phase, as might have been predicted from a consideration of classical disturbance hypotheses (eg Huston, 1979). Rather, there is a net decline in measures of community structure, in proportion to the quantity of waste present;

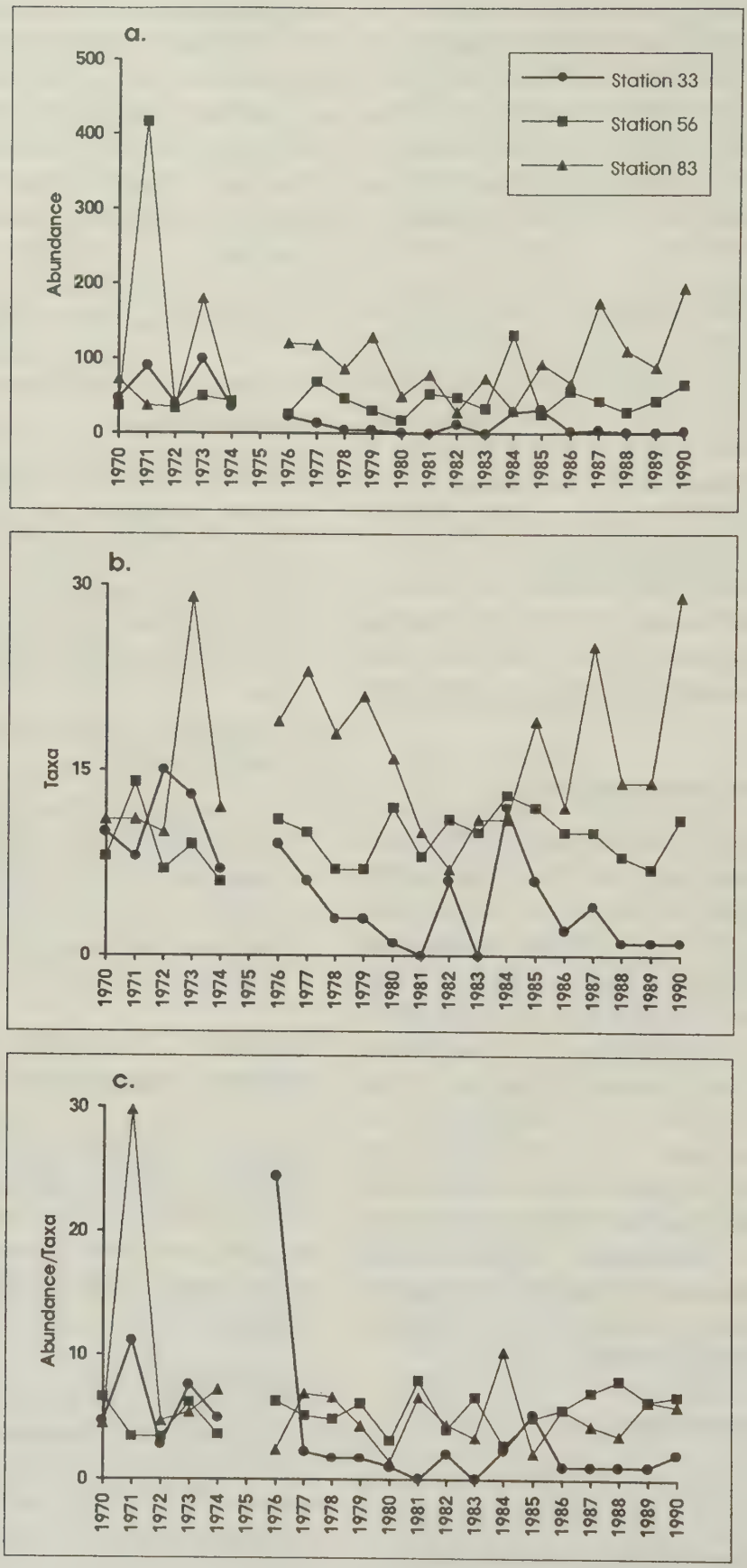
ii) the response of the benthos to organically-rich fine particulates associated with dredged material is, as might be expected, similar in kind to that occurring in response to sewage discharges or sludge disposal.

Biological Impacts Arising from the Marine Disposal of Dredged Clay

Dredged clay tends to occur in two forms, depending on the nature of the arisings and dredging techniques used. These, along with the effects of their disposal, can be illustrated by two case studies from Belfast Harbour, Northern Ireland.

Consolidated clay

In 1986, approximately 1 million tonnes of dredged clay arising from capital works in Belfast Harbour, Northern Ireland, was disposed of at the North Channel sewage-sludge disposal site. This material consisted of a red consolidated clay and the sea bed at the disposal area was apparently covered with substantial clumps (>100 cm diameter) of this material. During subsequent grab sampling of this area in succeeding years,



Figures 5a-c. Trends in abundance (a), numbers of taxa (b) and the ratio of the two (c) for the macrofauna at three stations in the vicinity of a solid waste discharge off the NE coast of England (numbers.0.1m^2; see text).

grabs have been recovered containing clumps of this material of apparently decreasing size. The infauna in this area did not appear to be affected and there was no evidence that the clumps had become colonised by benthic invertebrates either from above or below (this area was characterised by strong tidal currents (>120 $cm.s^{-1}$)); this may have been due to constant erosion of the clay surface preventing settlement and successful colonisation by invertebrates. In a series of studies, Maurer et al. (1981a, b; 1982) examined the recolonisation of a variety of dredged material types by a range of taxa. Although a number of external factors influenced recolonisation, from these studies it appeared that silt/clay was the least successfully recolonised. In the case outlined above, where clumps of clay are deposited on the sea bed, overall impact will probably be slight and presumably cause only slight displacement of the fauna due to mechanical effects on the area directly in contact with the material. The slow release of suspended silt/clay due to "weathering" is unlikely to be significant.

As a contrast to the above, large quantities of peat were disposed of at an adjacent location in the same year. After seven years, this material was still being recovered in bottom trawls and appeared to be more resistant to weathering as the clumps did not appear to be decreasing in size. In this case colonisation by invertebrates had occurred. Crevices on the exterior surface contained the bivalve Hiatella arctica; all the clumps examined to date were extensively bored by another bivalve Barnea candida.

Unconsolidated clay

The problems associated with the disposal of unconsolidated silt/clay material from Belfast Harbour have recently been assessed (Moorehead and McRobert, 1990). This assessment showed that the immediate effects of the disposal of such material are unlikely to be local unless the disposal site is extremely shallow with little water movement. Such areas are liable to have muddy bottoms and the organisms found may be expected to be relatively insensitive to the increased sediment load from the disposal of unconsolidated clay. Such material disposed of in higher energy systems may, however, be transported to more distant areas and the far-field effects become more relevant. These may have direct visual effects such as an increase in turbidity or the smothering of kelp and other epifaunal organisms. In a study of the impact of suspended clay to a range of macrofauna, McFarland and Peddicord (1980) found that organisms from muddy bottoms were insensitive to high suspended clay concentrations while open water fish and sandy bottom epifauna were more sensitive. In many cases it appears likely that the first indication of such far-field effects may come from recreational divers reporting increased turbidity.

In general, from the work of McFarland and Peddicord (1980) and Maurer et al. (1981a, b; 1982) it appears that, where the substratum resembles the material disposed of, elements of the local fauna will more rapidly commence recolonisation. In the peat-type material described above, other opportunists may begin to colonise the newly-available substratum.

Biological Impacts Arising from the Marine Disposal of Contaminated Silts

In the UK, the highest levels of persistent contaminants in dredgings arise from ports situated on estuaries with industrialised urban catchments (Murray and Norton, 1979), notably the Tyne, Tees and Mersey estuaries, which together contributed about 5 million tonnes of dredgings to the coastal seas in 1991. However higher levels of metal contamination in sediments have been studied in non-dredged areas affected by metalliferous minewater discharges (Burt et al., 1992).

In such areas, contaminant inputs are usually associated with high organic loadings in the dredgings, arising from industry and domestic sewage, potentially masking directly observable toxic effects of the contaminants. Benthic biological community response to

these dredgings, as assessed by standard benthic invertebrate monitoring methods (Rees et al., 1990), is usually similar to that predicted for sewage-sludge disposal (Rees et al., 1992) and can be regarded principally as indicative of organic enrichment and of physical perturbation due to changes in surface sediment granulometry. Even if direct impacts of toxic contaminants on community structure tend to be insignificant in comparison, the potential for bioaccumulation and transfer through the food chain provides a cause for concern and a justification for attempting to assess the biological significance of inputs of these pollutants in dredged material. However, as few dredgings disposal grounds for estuarine ports are remote from riverine input to coastal waters, it is often difficult to distinguish between the different contaminant pathways and to assess the relative importance of dredgings disposal compared with other local sources of these contaminants.

The composition of the dredged material is of crucial importance in determining the bioavailability of these contaminants. As interstitial dissolved oxygen levels are critical in determining partition equilibrium coefficients between sediments and interstitial water (Di Toro et al., 1990), the biodegradable organic content of the dredged material is an important factor. Reduced levels of oxygen in the interstitial water will mobilise metal contaminants, although complete anaerobiosis will immobilise many metals as relatively insoluble sulphides.

Other components of the dredged material, such as clay, may reduce the availability of toxins, particularly metals, due to adsorption of the contaminants onto the surface of the sediment particles. Attempts have been made to measure bioavailable metals in sediments using weak acid extractions, with some success (Burt et al., 1992).

Rowlatt and Webster (1992) proposed, as an initial approach, that action levels for contaminants in sediment should vary in proportion to the total organic carbon (TOC) content of the sediment, although this requires further testing for highly organic silts. However, it is accepted that other components of the sediment influence partitioning of metals between the silt and interstitial water and that uptake of toxicants by sediment-feeding biota will not depend solely on contaminant concentrations in the water phase.

Long (1992) summarised various approaches to the assessment of biological effects used in the USA to determine potential biological effects, including the use of Apparent Effects Thresholds (AET). This involves assessment of data on direct effects of individual contaminants on a wide range of biota and determination of the concentration above which statistically significant biological effects always occur. This then allows establishment of sediment contaminant concentration standards. Further details of proposed AET are given in Long and Morgan (1990), who report that clear AET values could be set for Hg, Cd, As, Cu, Pb, PCB, anthracene, chrysene and fluoranthrene but for other toxins, such as Cr, DDT and drins, no clear picture emerged.

Methods for direct measurement of biological effects of toxicants are currently under development and testing (eg Bayne et al., eds, 1988). As well as standard toxicity tests, these include metallothionein induction as an indicator of heavy metals, Scope for Growth measurements, which have been shown to be responsive to PAH and PCB contamination, and enzyme assay methods. (These methods are being studied by another GCSDM Task Team and will not be discussed further here, except to point out their potential value in identifying toxic effects in the presence of other masking influences.)

It should be stressed that direct methods of biological effects measurement have the advantage over analysis of concentrations of individual sediment contaminants, in providing a means of assessment of synergistic and cumulative effects of different toxins in a mixture.

Finally, measurement of bioaccumulation of toxicants is important in view of known impacts of organochlorine compounds on top predators and limits on levels of toxins in marine organisms for human consumption.

Chapman (1989) reviewed various approaches to the development of criteria for sediment quality, including bioassay and AET approaches. The conclusion of most studies appears to be that assessment of the biological effect of contaminants entering the sea through disposal of dredged material requires concurrent use of a range of techniques, including direct measurement of bioavailable toxins in the sediment, taking account of TOC and physical perturbation, coupled with measurement of levels of bioaccumulation in the fauna (and possibly flora for nearshore disposal sites) and biological effects techniques.

Few data using direct biological effects measurements are yet available for UK dredgings disposal sites and studies involving only sediment analysis and benthic community studies have typically been unable to isolate effects of toxicants from other effects, such as organic enrichment, as described by Rees et al. (1992) for Liverpool Bay. However, the potential for the use of community-level responses in the field in order to distinguish between the effects of specific contaminants has recently been demonstrated in the laboratory using cultures of sediment-dwelling meiofauna dosed with a variety of concentrations of heavy metals (Austen et al., in press).

Biological Impacts Arising from the Marine Disposal of Dredged Muddy Sands

Muddy sands may arise from capital (ie "one-off") dredging projects, but maintenance dredging is a much more common source and continuous, even daily, inputs may be the normal practice, especially in the vicinity of major ports.

The contaminant burden in dredged material is predominantly associated with the fine sediment particles. Rowlatt (1988) has suggested that the "fines" content of bottom sediments in inner Liverpool Bay and the Mersey are represented by a single population, and hence it is the quantity of these present, rather than concentrations of associated contaminants, which determine burden.

At disposal sites the fine component of natural muddy sand sediments confers at least some degree of short-term stability through its cohesive properties (in contrast to well-sorted fine sands which may be more regularly disturbed, eg by tidal action). Such sediments are therefore likely to support a moderate to high biomass of macrofauna; however, they may be periodically vulnerable to disturbance/wash-out from storm effects.

Examples of muddy sands receiving sedimentologically similar maintenance dredgings include locations in Liverpool Bay (see, eg Norton et al., 1984; Rees et al., 1992) and off the Tyne (see, eg Eagle et al., 1979; Rees and Rowlatt, 1994a). The former is located in shallow water of about 10 m, and sediments of this nature in the general area typically support a productive fauna. Besides dredging inputs, other anthropogenic influences may arise through discharges from the Mersey estuary and offshore disposal of sewage sludge, while periodic storm disturbance represents a significant natural

Table I Major macrofauna taxa from muddy sand locations

	Liverpool Bay (1 mm mesh)			**Tyne** (1 mm mesh)	
		Nos 0.1 m^{-2}			Nos 0.1 m^{-2}
1.	Mysella bidentata	121	1.	Lumbrinereis sp.	45
2.	Lanice conchilega	115	2.	Prionospio sp.	36
3.	Lagis koreni	88	3.	Thyasira flexuosa	26
4.	Abra alba	32	4.	Nucula nitidosa	17
5.	Nucula nitidosa	31	5.	Ophiura sp.	14

influence. The Tyne site is located in deeper water (about 30 m) and the sediments generally support a lower biomass, and are more stable. The fauna is also potentially influenced by Tyne outflow, along with other marine disposal activities, including colliery waste, fly ash and sewage sludge. (The risk of confounding effects of dredgings disposal with other anthropogenic influences, especially at some distance from the activity, should always be considered at the outset of any investigation: see Survey and Monitoring Section below).

The top five ranked macrofaunal taxa from representative muddy sand locations in Liverpool Bay and off the Tyne are shown in Table I

These data amply demonstrate the very different character of the two assemblages. For example, only one species (Nucula) is common to both. Such contrasts in the community structure (and biomass) of muddy sands are to be anticipated at a number of other disposal site locations.

Impacts at varying distances from a regularly used disposal site may take the following forms:

i) Immediately beneath the discharge, smothering of most species will occur, though some of the more robust ones may have a capacity to migrate upward if not too deeply buried (see, eg Maurer et al., 1981a,b; 1982); the introduction of "exotics", eg estuarine species which have survived the process of dredging and disposal, is also a possibility (see, eg Wildish and Thomas, 1985). Clearly, in the immediate aftermath of disposal, the surface of newly-deposited sediment will generally be devoid of indigenous macrofauna;

ii) Adjacent to the discharge (within the disposal site), the fauna may be affected by high turbidity and dispersing fine material moving as bed-load; the possible influence of particle-bound contaminants and organic matter must also be considered (the latter providing a potential source of food, or having a role in the development of anoxia within sediments if present in sufficient quantity). The most tolerant/adaptable species will be favoured. High suspended-solid load may favour deposit-feeders over filter-feeders; such sediments would typically support a faunal community at an early successional stage but, in inshore exposed areas, such a community may not differ appreciably in terms of the range of taxa from that of unaffected sediments nearby. It is conceivable that deposited material will confer temporary stability in a high-energy area, thus promoting colonisation by certain species (Rees et al., 1992). As a result, benthic productivity may be enhanced above background levels, which could be viewed as a beneficial consequence of dredgings disposal, assuming that there are no risks of contaminant bioaccumulation and trophic amplification (see eg Rhoads et al., 1978). Recolonisation may occur both through normal larval recruitment patterns and migration of adults from nearby populations in such areas;

iii) On the outer fringes of a disposal site, assuming these are rarely if ever in direct receipt of dredgings, a more "stable" community may develop, comparable to nearby unaffected locations, but modified due to the influence of dispersing material;

iv) Beyond the disposal site, but still within the "sphere of influence" of dispersing particulates, effects may include a net enhancement of benthos due to the stabilising or nutritional properties of settling material (eg Rees et al., 1992; Zambriborsch et al., 1982). The spatial extent of this zone, and the severity of impacts within it, will depend on the dispersive properties of the receiving area, and the quantities disposed of. Effects on the benthos specified under ii. above may extend beyond the disposal site boundary in some circumstances.

Indicator organisms — characterised either by their presence or absence —will tend to vary in nature according to local

environmental conditions. However, accumulations of organically-rich sediments might be expected to support populations of "classical" opportunists such as certain Capitellid or oligochaete worms (see Pearson and Rosenberg, 1978). Gray (pers. comm.) suggested that polychaete worms of the family Spionidae are indicative of disturbance though not necessarily of organic enrichment.

Inward-migration of mobile epibenthic predators/scavengers, including echinoderms, crabs, shrimps and fish may occur. This may be to exploit any enhanced biomass peripheral to disposal sites, or in response to the presence of dead or moribund individuals of certain species newly affected by dispersing material within or very near to the disposal site.

Within the disposal site boundary, there is a strong likelihood that deposited material will initially be present as discrete mounds rather than as a uniform "blanket". At a regularly used site, the consequence for the fauna of patchy impacts spread over time is likely to be the production of a mosaic of different stages of development in the recolonisation process, ranging from near-azoic to near-normal, and hence sampling is likely to reveal a high degree of background variability.

Clearly, the chemical quality of the dredgings may have a significant bearing on observed changes; interactions between physical and chemical effects may therefore complicate interpretation of changes in the field.

Biological Impacts Arising from the Marine Disposal of Dredged Sands

There is relatively little published information regarding the disposal of dredged sand to sandy areas. In 1991, a benthic survey of the region adjacent to Sizewell Generating Stations A and B off the English east coast was carried out for Nuclear Electric plc (IECS, 1992; see also Fig. 6). A sublittoral site within the study area was used for the temporary storage of the fine sands necessary for construction. The impact of this storage (which lasted less than one year) was assessed.

The study area supported a community largely characteristic of mobile medium and coarse sands. It was therefore considered that dredgings disposal, or sea bed storage, of material similar in nature to the indigenous sediment would have had a minimum effect on the coarse areas. The disposal of finer material might have had the effect of smothering the sea bed fauna but it was predicted that, in such a high-energy environment, the finer material would be rapidly dispersed, with a nearby muddy trench (which was dominated by the bivalve molluscs Macoma and Nucula, and the heart urchin Echinocardium) being the likely "sink" for some of this material.

Species of the amphipod genus Bathyporeia were present at sandier locations to the south, with lowest abundances surrounding the dredgings disposal site. The genera Magelona (polychaete worm) and Fabulina (bivalve mollusc) are typical of mobile sandy sediments. In the study area, the former was most abundant at a station on the offshore side of the disposal site; Fabulina occurred at stations in the offshore sand bars, although not in the disposal areas. Following the application of various statistical methods, it was concluded that the stations surrounding the dredgings disposal area were unaffected by the disposal operation. Two of those stations were relatively poor, in common with the inshore stations, whereas another two supported a rich fauna in common with the stations at the southern part of the deeper muddy trench. However, one station adjacent to the disposal site had a mixed assemblage which combined elements suited to coarser, mobile sand areas (amphipods and Magelona) with those more commonly associated with finer, more organic sediments as found in the muddy trench area (Nucula, Macoma and Echinocardium).

The mostly "high energy" nature of the present study area, together with the short duration (<1 year) of disposal, would have led to rapid recolonisation and hence a rapid rate of sea bed recovery. The present study indicated that the area surrounding the temporary dredgings disposal site had a "transitional" fauna that represented a

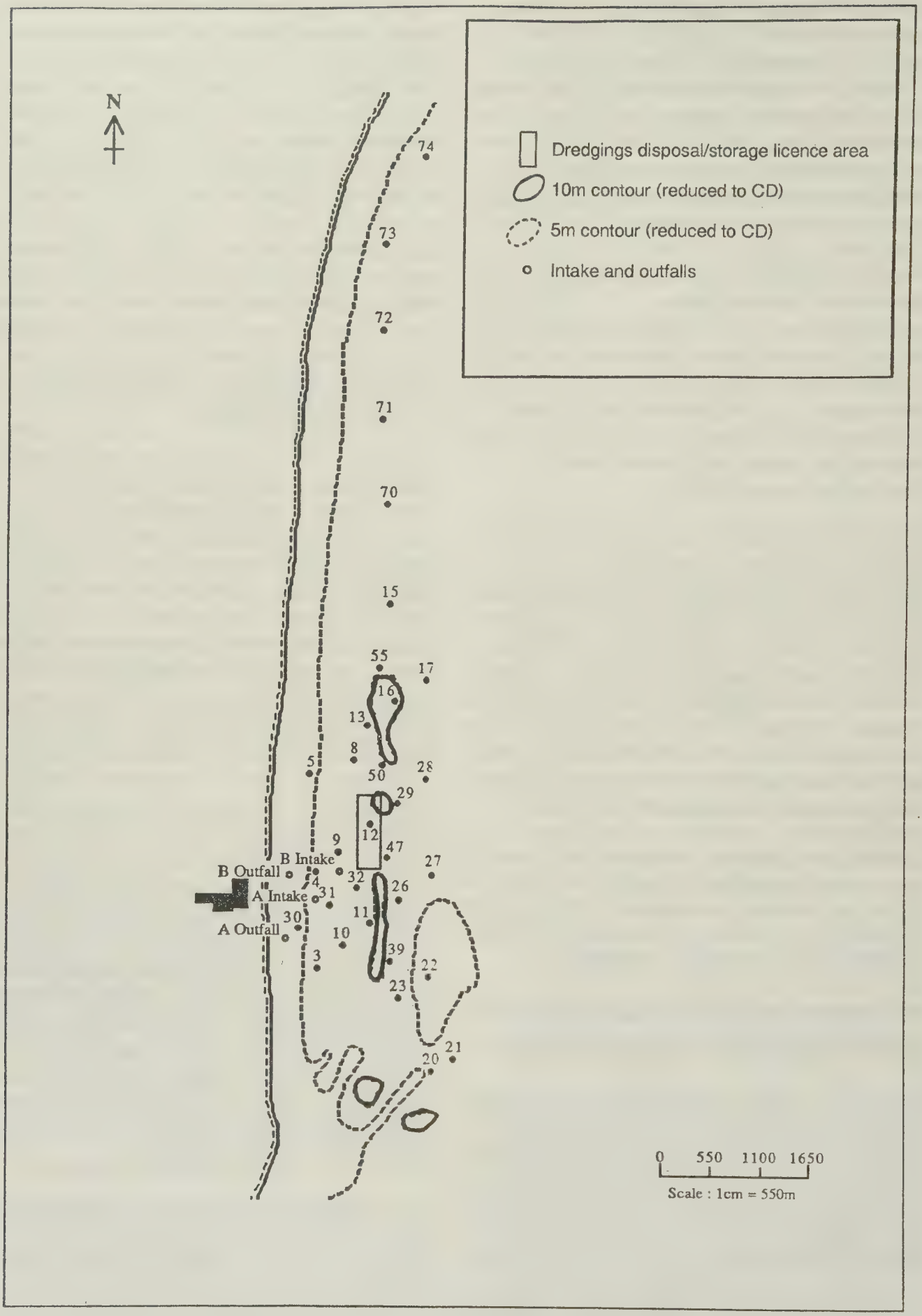


Figure 6 *Station positions for a benthic survey conducted in 1991 around the Sizewell nuclear power station, eastern England.*

boundary between different communities. Dredged material that was finer than sediments in the receiving area would, under the prevailing hydrographic regime, have been transported to a lower energy area such as the muddy subtidal trench. The transport and deposition of that material is unlikely to have resulted in a change to the fauna in that area as the taxa encountered were adapted to "depositional" conditions. In this case, as the disposed/stored material, resulting from the dredged culvert for the intake and outflow channels, was largely similar to or coarser than the receiving area (Nuclear Electric: pers. comm.) then most will have been redistributed within the "high energy" environment.

It may be concluded that the disposal of dredgings at the temporary disposal site has had a minimal effect. This was to be expected given the "high energy" nature of the area, the associated benthic communities and the nature of the dredged material.

Other studies show that the benthic effects of the disposal of "inert" solids such as clean sands similar to those deposited on the temporary dredgings disposal site off Sizewell, are poorly understood. Unlike the effects of organic enrichment, there is no widely accepted model of the changes to be expected. In the immediate vicinity of disposal, the primary effects will be due to smothering of the indigenous fauna and the structural and chemical changes to the sediment resulting from the incorporation of material finer than that present previously (Oslo Commission, 1993). As a result, heavily-used dredgings disposal sites will generally support a lower biomass and abundance than the adjacent areas (Wildish and Thomas, 1985; Zambriborsch et al., 1982). The magnitude of these changes will clearly depend upon the nature of the material and of the receiving environment (eg Probert, 1975; Bamber, 1984). In "high energy" environments, colonising organisms will be those able to tolerate regular or intermittent disturbance of the sea bed (eg Magelona, Lagis and certain amphipods); the rate of recolonisation of newly deposited sediments will depend on the nature of the surrounding fauna and the time of year in relation to seasonal recruitment patterns. The effects of disposal in such areas are likely to be relatively short-lived and recovery relatively rapid after the cessation of dredgings disposal (Oslo Commission, 1993).

An alternative approach to understanding the underlying mechanisms of response to dredgings disposal is the use of experimental laboratory systems (eg Maurer et al., 1981a,b; 1982). These mechanisms include selective migration of adults from outside the receiving area as well as vertically through recently deposited dredged material. Opportunistic polychaetes and small mobile crustaceans (eg cumaceans and amphipods) were the initial colonisers. Although all small crustaceans are killed by heavy deposition (especially of sand), certain amphipods (particularly haustoriids) and molluscs can rapidly migrate and recolonise. Maurer et al. (op. cit.) also considered that temperature will affect recolonisation through its effect on burrowing speed. Similarly, the seasonal timing of disposal in relation to natural recruitment cycles will determine the rate of recolonisation.

It can be concluded that:

i) Published information on sand disposal to sandy (or other) areas is limited; nevertheless, available evidence suggests that sand disposal to "high energy" sandy areas will generally have minimal effects;

ii) In such "high energy" areas, recolonisation will be primarily by molluscs, small crustaceans (especially haustoriid amphipods and cumaceans) and polychaetes adapted to mobile sediments;

iii) The rate of recovery following cessation of disposal will be relatively rapid (possibly of the order of months);

iv) Rates of recolonisation/recovery will be affected by the timing of a "once-off" disposal operation; if it immediately precedes seasonal peaks of recruitment of juveniles from the plankton, the rates may be very rapid;

v) laboratory-scale experiments may be a useful adjunct to field studies in improving understanding of the mechanisms of benthic recolonisation.

Biological Impacts Arising from the Marine Disposal of Dredged Coarse Materials

A wide variety of coarse materials may be removed to sea for disposal at offshore disposal grounds. These materials may have the following origins.

Rock

Rock comes mainly from capital projects that will frequently have involved blasting. Examples of this type of capital dredging work in UK waters have included the removal of rocky outcrops obstructing harbour entrance channels (eg Milford Haven, Wales) and where pipelines have to be brought through steep rocky coastlines (eg Amlwch oil terminal, North Wales).

The total amount of material for which sea disposal is likely to be needed will be fairly small by comparison with sand or mud dredgings. In coastal projects there will frequently be other uses for large rocks. Only where the rock is very soft (such as in the case of chalk or Tertiary sandstones) is it possible that moderately large amounts might need to be disposed of to sea.

Rocks from operations involving blasting will be of a wide range of sizes and will be sharply angular when first disposed of. They will usually have been picked up after blasting by grab bucket or sometimes ladder bucket dredgers. A certain amount of fine rock flour giving rise to slight turbidity plumes is liable to be released as well.

Boulders and cobbles

Much of the seabed off British coasts north of the Severn-Wash line has glacial till overlying the bedrock. Reworking of this has in many places left the actual bed covered by a veneer of lag including boulders and cobbles. Capital dredging projects for ports (as well as trenching for pipelines) frequently result in the requirement to excavate glacial boulder clay, so that considerable quantities of boulders and cobbles need to be disposed of. The enlarging of the channel from Barrow docks (NW England) is a prime example of a major capital project that has cut into glacial till.

In the short term, a distinction can be made between boulders that have been exposed at the bed surface and those that are embedded in a clay matrix. The exposed ones will already be colonised by epifauna or flora, some of which may survive the dredging process.

Off the southern coasts of Britain there are also many places where boulders and cobbles, including flints, lie on the seabed as relics of the erosion of former beach platforms. Capital works may sometimes require the disposal of such material though, given the shortage of aggregate in southern Britain, disposals are likely to be limited.

Pebbles and gravels

Sources of pebbles will be from dredging of boulder clay, from outwash gravels in the glaciated part of the country, and coastal shingle banks.

Anthropogenic coarse material

This material includes clinker, slag, concrete, dressed stone, brick, slates, tiles and numerous other sorts of port refuse and wreckage. In the past, much material of this sort was taken to sea, for example, to Liverpool Bay during the clearing of war-damaged buildings. Nearly all harbour dredging close to quay walls will bring up extraneous items of solid refuse: tyres used as fenders or discarded lengths of wire cables may be particularly frequent. In such cases, licence conditions may be imposed which require screening for larger artefacts, which are then removed to land for disposal.

Biogenic coarse material

Around much of Britain the complex fluctuations in sea level since the last glaciation resulted in the frequent presence of beds of peat and submerged forest several

metres below present sea-levels. In consequence, works to enlarge harbours have frequently required the dredging of biogenic material of this sort. To this may be added ancient waterlogged timbers including those from very old wrecks, harbour works, ancient trackways and fish weirs that are barely distinguishable from natural submerged forest.

There may, exceptionally, be developments requiring the "capital" dredging of calcareous deposits such as maerl, eg along parts of the south-west coast of England. There have also been instances where the bones of large pre-historic animals including mammoths have been brought up in capital dredging projects.

Refuse

Up to 35 years ago, it was the practice at several port cities to barge refuse to sea and, even now, such disposal grounds still have a lot of ash in the surface sediments. Broken glass and pottery is also still very frequent at such sites. Historic disposal grounds and anchorages can often be expected to have such material.

Behaviour of dredged material following disposal

By definition, most coarse materials disposed of to sea will have properties making them non-dispersive. Most will go straight to the bottom and will be too large and dense for movement by tidal currents. Any dispersion at all from the deposited mounds of a particular barge load will often depend more on intermittent storm waves than on tidal currents. The scatter will also be much influenced by the erosion/dispersion of other finer materials in the mounds.

Dispersion of the actual coarse material, as opposed to clays associated with it, will be slight. In most cases, coarse material will stay permanently within a very small radius of where it is actually deposited. After disposal, there will be periods of months or years during which erosion and wave action breaks down the integrity of barge loads and exposes stones hidden in a clay matrix. This process will be erratic. The hard material most likely to be moved significant distances from a disposal site will be peat. Peat lumps tend to wear so that they roll on the seabed. Clay lumps can also behave in the same way in strong tidal current situations. The action of fishing operations in dispersing solid debris from disposal grounds must also be borne in mind.

Where stones and other lumps of solid material are deposited individually onto sandy ground, they may develop small current scour pits around themselves. The net result is that such stones are inclined to sink into the sediment to the level where they meet a harder basement. Deposited stones can be expected to emerge intermittently onto the surface of a sand bed as banks, sand waves or mega-ripples move.

Ecological effects

Unless they have a friable and continuously eroding surface like balls of clay, rocks, boulders and cobbles will very quickly be colonised by a range of epifaunal organisms. In the short term, something approaching an ecological succession is to be expected as a fouling epifauna colonises the new hard surfaces. The range of species will depend on the location and depth of the disposal site and the nature of the material. On sandy grounds hard materials may be subjected to sand-scour affecting the surfaces. In turbid water areas, the settlement of fines will partially inhibit recruitment. Boulders deposited shallow enough in the sublittoral to be in the photic zone will be colonised by macro-algae as well as typical epifaunal species.

There have been few studies where the colonisation of deposited rocks has been investigated. However, parallels can be drawn from studies of artificial reefs and rocks that have been placed as "rip-rap" in coastal defences. One dredgings disposal operation that was easily observed was associated with the laying of a pipeline across the Menai Straits (N Wales). The trench for this pipeline was dredged through glacial till using a ladder bucket dredger and the material was

disposed of by barge in water so shallow that on extreme spring tides the mounds representing individual barge loads were just uncovered. The boulders that were brought up with the glacial till were colonised within a few months with a dense cover of Balanus crenatus, with a small amount of weed on the tops. Figure 7 shows a diagrammatic section through one of these mounds about 6 months after disposal. Although almost into the littoral, it would be expected that much the same would apply to the disposal of boulder clay on a stable sandy ground offshore.

Boulders of calcareous rock including even hard Carboniferous limestone can be expected to be colonised in time by borers. Boring results in secondary diversification of the habitat. It is very common when sampling in areas of coarser ground in the Irish Sea that the limestone cobbles have much more fauna on them than igneous ones of the same sizes.

It is possible that where heavily mineralised metamorphic rocks are being disposed of after being blasted, then there will be an initial delay in colonisation while the new surfaces weather.

Dredged material is frequently characterised by having particle size compositions that are different from the native material at the disposal sites. In the case of coarse material deposited onto finer sediments, the effect will frequently be to diversify the available micro-habitats. This will be particularly pronounced when calcareous rocks, easily pitted by the actions of boring organisms, are disposed of to sea. The contorted and pitted surface of clinker is also liable to be colonised by a particularly wide range of crevice-seeking species.

Through the diversification of the habitat, stable coarse material will therefore usually result in an increase in the total number of benthic species within a disposal ground. In this respect, there will be parallels with the way artificial reefs increase the diversity of species in an area. However, in most cases the coarse material will only be one component of a mixture of types of material.

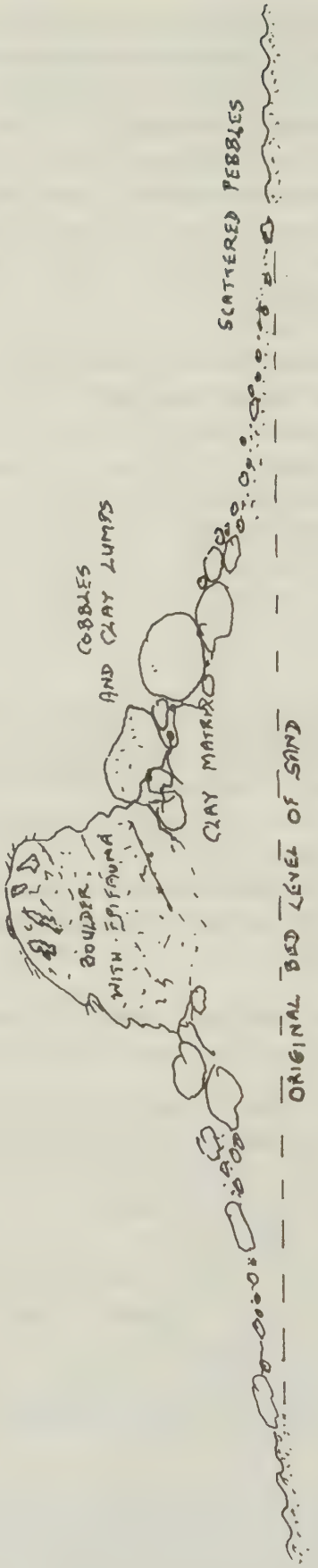


Figure 7. *Diagrammatic section through a mound of boulder-clay dredged material about 6 months after disposal on the edge of a sand bank at the entrance to the Menai Straits. The mound, standing about 1.5m above the natural sand level, represents the residue of a single barge load. Dredging was by ladder bucket dredger being used to excavate a pipeline trench.*

In the immediate vicinity of disposal, the net effect in terms of diversity will be a balance between the depressing effects of blanketing and turbidity, and the enhanced habitat complexity of the coarse materials.

Generic Hypotheses for Biological Effects of Dredgings Disposal

Published studies on the biological effects of dredgings disposal around the United Kingdom and elsewhere in European waters are limited. A large amount of effort in the USA has been put into the development of methods for the management and monitoring of dredgings disposal sites, but most reports arising from this effort appear in the "grey" literature.

Peddicord (1987) provides three basic (but useful) conclusions concerning approaches to monitoring at dredgings disposal sites, as follows:

i) Dredged material is sediment and behaves environmentally like sediment, unless it is disposed under conditions that alter its geochemistry;

ii) Evaluative procedures technically appropriate to in-place sediment may be used for dredged material deposited at aquatic disposal sites;

iii) the potential environmental impacts of dredged material disposal can be accurately evaluated using procedures that consider its geochemistry and the physico-chemical and biological conditions at the disposal site.

These statements provide an equally valid framework for UK activities. It is important to note the emphasis placed (in iii.) on the requirement for site-specific data in impact assessments. To this may also be added the importance of a knowledge of the physical structure of dredged material, eg as may be revealed by particle size analyses.

One consequence of the limited available literature concerning effects on the benthos of dredgings disposal is the difficulty it creates for the establishment of descriptive "models" which are firmly grounded in empirical data. However, it could be argued that, even with a larger UK database, information reflecting the high degree of environmental variability between disposal site locations, along with disposal site practises, would not easily lend itself to the sort of generalisations that may be permissible for, say, the effects of organic waste inputs.

One important distinction to be drawn in assessments of the biological effects of dredgings disposal is between conditions prevailing at a regularly-used disposal site, where management interests will generally lie in ensuring spatial limitation of effects, and those at a site used only very irregularly, where interests may also lie in the rates of physical and biological "recovery".

Recurring Features and Concerns about the Biological Effects of Dredgings Disposal

It may be anticipated that there will be a shift in the nature of the benthic fauna recolonising newly deposited material towards that naturally associated with substrates of similar character. Factors which might complicate this generic assertion notably include variability in the "quality" of the dredged material, and differences in environmental conditions between those at the disposal site and those at locations naturally supporting comparable sediment types.

In "high energy" environments (eg one supporting mobile sands), the disposal of cohesive muddy or coarse sediments may confer a degree of temporary or longer-term stability in the receiving area, and hence increase the potential for recolonisation by benthic organisms (see eg Rees et al., 1992).

Individuals of certain benthic species which survive the process of dredging of superficial deposits may also survive the process of disposal; this can lead to the local occurrence of "exotics", ie species which would not normally be associated with the prevailing habitat type in the disposal area (eg Rees et al., 1992; Wildish and Thomas, 1985).

Enhanced growth and/or survival of "opportunistic" benthic species in the vicinity of dredgings disposal sites may be considered to be a beneficial consequence, eg by increasing the availability of food for fish (see Rhoads et al., 1978). Such enhancement, if present near to a regularly used disposal site, will be sustained and therefore of much greater significance than that arising from a "once-off" disposal operation. However, the benefit to local fish populations may be offset by the risk of bioaccumulation and transfer through the food chain of any contaminants associated with the deposited material (see eg Rees and Rowlatt, 1994b).

Primary toxic effects of dredged material should not (in principle) be a problem, as chemical analyses and other screening procedures, eg sediment bioassays, are designed to prevent the sea disposal of unacceptably contaminated material (see Oslo Commission, 1993); however, secondary toxic effects may arise from the development of anoxia within or beneath deposited sediments, notably those enriched in organic matter. Such effects can be described by the model of Pearson and Rosenberg (1978), referred to below.

The behaviour of "plumes" of finer sediment generated during disposal operations merits further study in UK waters, especially where there is a risk of "far field" accumulation of fine particulates in sensitive coastal areas. As well as any adverse consequences for the fauna arising from the settling in quantity of such material, sustained increases in water-column turbidity might be expected to inhibit the survival of suspension-feeding organisms.

Descriptive Summaries of Benthic Responses

The disposal of coarse material such as boulders and rock is more likely to arise from "once-off" capital projects, and hence re-colonisation is envisaged to follow classical successional events analogous to those on rocky shores (Fig. 8a). Such coarse material, if left relatively undisturbed, is also likely to be favoured by fish and shellfish, either as a source of food or as refuge, and adults of these groups may arrive very rapidly through inwards-migration after disposal. An analogy may therefore also be drawn between the disposal of coarse dredged material to a site characterised by the natural occurrence of finer sediments, and the construction of artificial reefs to enhance commercial fish or shellfish populations (see, eg, Woodhead et al., 1985).

Responses of the benthic fauna to deposited material which is enriched in organic matter (eg arising from the maintenance dredging of muddy depositional areas in an urbanised estuary) may be expected to conform with the empirical model of Pearson and Rosenberg (1978). Thus, along a gradient of increasing organic enrichment as in Figure 8b, an initial enhancement in the benthic community is followed by a structural change as a few small-sized "tolerant" species, present in high densities, are favoured over a wider range of larger-sized species which are more sensitive to degradative changes within sediments.

The experience of Rhoads et al. (1978) might suggest that newly deposited sands (or other "clean" soft sediments) would be preferentially re-colonised by "opportunistic" benthic organisms which thrive in disturbed habitats, giving enhanced abundances compared with nearby. While this may be a feature of certain localities (perhaps especially within estuaries), there was no evidence for enhancement in response to the disposal of "inert" materials such as china clay waste (Probert, 1975), fly ash (Bamber, 1984) or fine particulate waste from a potash mine (Craig and Shillabeer, 1993). Rather, the pattern in these cases was one of linear decline in the indigenous fauna in proportion to the amount of waste present. Neither was there any evidence for the occurrence of "pollution indicator" species at high disposal rates. Similar observations have been made at mixed solid waste disposal sites (ie colliery waste, fly ash and dredgings) off the north east coast of England (Eagle et al., 1979; Rees and Rowlatt, 1994a). Present evidence would therefore suggest that a straightforward linear model (Fig. 8c) would best describe the effects of the marine disposal of "inert" (soft

sediment) dredgings. In relatively quiescent areas, the time-scale for recovery may be measured in years, rather than months, as the material may have little attraction for microbial or larger growth forms until processes of weathering and incorporation into natural sediments have reached an advanced stage.

Disposal of individual loads of dredged material generally takes place instantaneously through bottom discharge and also at discrete time intervals, although these intervals may be short at heavily used sites. One result of this will often be the presence of discrete mounds of material on the sea bed. At any one time, these may be eroded and/or recolonised to different degrees depending on age and their proximity to freshly discharged material. Sampling in such areas is likely to reveal a mosaic of different developmental states in the benthic fauna, and the data are therefore likely to be a good deal more variable than suggested by the summaries of Figure 8a-c as the disposal area is approached.

Conclusions

Most dredged material, irrespective of its nature, will be rapidly re-colonised by benthic species following disposal. The rate of recolonisation will clearly depend on the frequency of disposal, and the nature of the receiving area. The timing of disposal in relation to seasonal peaks of recruitment of juveniles from the plankton will also affect re-colonisation rates. However, adults may arrive immediately, either through vertical migration if undamaged and only lightly buried, or horizontally (passively or actively) from unaffected areas nearby. Factors influencing time-scales for "recovery" on cessation of disposal include the dispersive properties of the receiving environment, the degree of similarity of the deposited substrate with that naturally prevailing in the area and the age composition of "key" species in unaffected populations.

The "models" of Figure 8a-c are presented as an aid to the formulation of hypotheses concerning the likely consequences of individual disposal operations. In some well-defined cases, these models may provide an acceptable description of actual events. However, it must be stressed that site-specific predictions of benthic changes (and hence the utility of the models) may be greatly complicated by changes in the nature of the source material to be deposited over time, its frequency of disposal and the presence of other inputs to the locality.

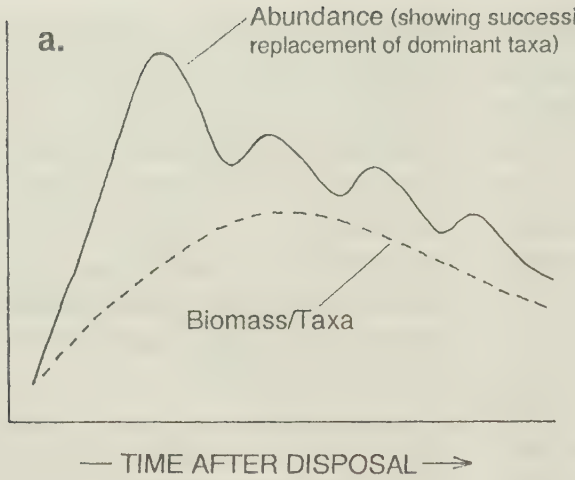
Derivation of "Environmental Quality Standards"

GCSDM specifications (see MAFF, 1993b) recognise the likelihood of adverse consequences at the sea bed within the disposal site itself, arising largely from the immediate settling of sediments following bulk discharge. The main objective at most existing disposal sites will therefore be to minimise the intensity and spatial scale of effects beyond this "mixing zone". However, for the disposal of "clean" dredged material to a highly turbulent area, the objective may be to maximise dispersion and hence minimise local physical effects by systematically spreading the discharge over a larger area.

The approach to sampling for EQS compliance will be similar in principle to that developed for sewage-sludge disposal sites (MAFF, 1993a), ie involving comparisons between "Treatment" and "Reference" sites, and may be summarised as follows :

EQS 1: monitoring for acceptability of change within the sphere of waste influence (but outside a "mixing zone", where defined).

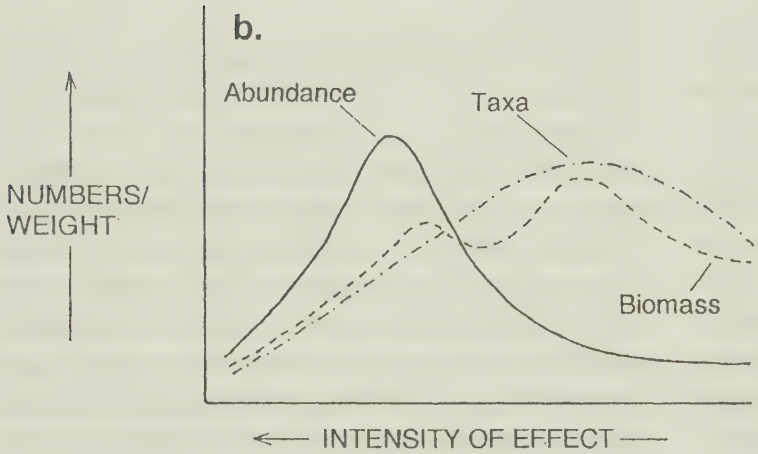
As outlined above, this will need to take account of site- and material-specific aspects of the disposal operation. Further, development of criteria for "acceptability" will depend upon locally-adapted impact hypotheses. An empirical model for benthos responses to organic enrichment is likely to be useful in this respect (see previous section), but will not account for all waste types encountered. It should also be remembered that some types of disposal activities may even have beneficial consequences for the receiving environment,



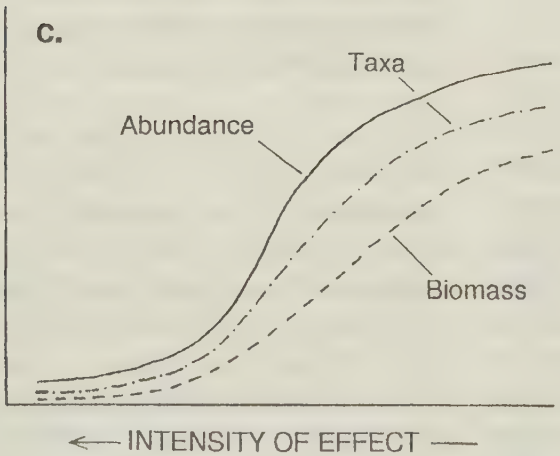
'ONCE-OFF' (CAPITAL)

Recolonisation --> 'climax' community

This may proceed very rapidly (within weeks or months) in 'high-energy' areas, depending on dredged material quality. In **a.**, it is assumed that sediments are coarse and stable in nature



'REGULAR' (MAINTENANCE) : ORGANICALLY-ENRICHED SEDIMENTS



'REGULAR' (MAINTENANCE) : 'INERT' SEDIMENTS

Figures 8a-c. Simplified models of changes in the benthos in response to the disposal of dredged material (b. after Pearson and Rosenberg, 1978)

Note 1: the y-axis scales for Abundance, Taxa and Biomass have been arbitrarily adjusted to facilitate assessment of relative change in these measures. The origin of the x-axis represents the most extreme manifestation of biological impact

Note 2: NB. In most cases, a 'mosiac' of successional states can be expected in the vicinity of the disposal site, associated with discrete patches of deposited material (see text)

for example, by enhancing benthic productivity.

Selected quantitative measures describing changes in benthic community structure will be as for sewage-sludge disposal sites. For the disposal of dredged material consisting largely of organically-enriched sediments, then boundaries for permissible variability in these measures will also be as for sewage-sludge disposal sites. However, alternative boundaries will have to be defined for other types of dredged material.

In the particular case of the disposal of coarse deposits which are largely invulnerable to water movement, then a monitoring programme, if required, may entail purely physical study (eg side-scan sonar and bathymetry) at the disposal site. However, there will be an interest in rates of recolonisation and recovery of the benthos.

EQS 2: monitoring for compliance with the status quo outside the sphere of waste influence.

This task should not present difficulties, provided that adequate "Reference" sites can be identified. A good knowledge of dispersal pathways will also be essential for locating stations.

Selected quantitative measures describing changes in benthic community structure, along with boundaries for permissible variability in these measures, will be as for sewage-sludge disposal sites.

Regular re-appraisals of sampling design will be necessary in order to ensure the continued effectiveness of monitoring programmes for EQS compliance. A critical factor in this respect will be any temporal variability in the nature and quantity of material disposed of to sea.

Survey and Sampling Strategy

Factors to be Considered in Deciding upon Biological Monitoring Approaches

Nature and origin of material

The consistency of the material (ie whether "hard", "soft" or "mixed") will largely determine the sampling techniques to be deployed at the disposal site. It will also influence judgements on the spatial scale of the design, and on the relative emphasis to be placed upon physical, chemical or biological assessment. (The practicalities of obtaining more precise information on sediment granulometry at the site of dredging needs to be considered; this may involve a combination of core profiling and superficial sampling, depending on anticipated vertical or horizontal variability).

The contaminant burden will similarly influence the scale and intensity of monitoring. Both factors may vary with the location of the source material, and the depth to which it is dredged. Sediments may, for example, originate from "high-energy" channel systems where they are subject to regular disturbance by tidal current action, or from more quiescent depositional areas. Especially within estuaries, proximity to waste discharges from urban or industrial developments will be relevant to later assessment of the consequences of dredgings disposal.

Nature of disposal site

The extent to which hydrodynamic conditions promote dispersion of deposited material will be critical to the design of benthic monitoring programmes. In highly dispersive areas in receipt of significant quantities of fine sediments, the possibility of "far-field" effects must be addressed.

Both the nature of the deposited material and that of the receiving area will determine the type of biological response to be expected, and its severity. They will also determine the bottom sampling methods to be used. For example, hard substrates will not be suitable for traditional grab samplers.

Another factor to be considered in sampling design includes the proximity of the site to commercial fish/shellfish stocks. In addition to the possibility of direct impacts, aspects

such as the role of the benthos as a food-source for higher trophic levels, and the potential for contaminant bio-accumulation may need to be addressed. Nature conservation and recreational interests will also have to be considered during monitoring and assessment of the consequences of disposal, as will other waste disposal activities nearby. (These matters are dealt with in the Oslo Commission guidelines for the management of dredged material : see below).

Nature and duration of disposal operation

These may vary widely from site to site. For example, disposal of maintenance dredgings through an annual licence may take place over a relatively short time-interval during an intensive "campaign", or alternatively may involve regular disposal of smaller amounts over an extended period of time. Equally, disposal activities at a site may in some cases be separated by several years (this may be especially true for dredgings from "capital" projects), or involve material from different sources which may vary greatly in character.

A case may be made for the disposal of "clean" dredged material within narrow boundaries in order to confine (gross) physical and biological effects to a small area. Alternatively, the aim may be to disperse the material over a much wider area in order to reduce the likelihood of any significant impacts occurring. An additional option which has been attempted in US waters is the "capping" of contaminated material after disposal with a clean overburden of sand (see eg Rhoads et al., 1978). Another approach aimed at limiting the persistence of biological effects and maximising the effective use of the natural dispersive properties of a receiving area may involve the alternating use of two disposal sites. The "fallow" period would allow for longer-term dispersion of deposited material along with substantial benthic recolonisation prior to re-use (see, eg, Kleef et al., 1992).

Clearly, a good knowledge of disposal practises will be very important in deciding upon the nature of the biological response to be anticipated, the amount of monitoring effort required and the frequency of sampling.

Magnitude of disposal operation

This is likely to be one of the most important factors determining the extent of monitoring at a site (if any).

Monitoring Strategy

Impact hypotheses

The development of "impact hypotheses" for disposal sites (see Oslo Commission, 1993) is strongly supported. The construction of generic "models" for the responses of the benthos to different categories of dredged material of the type outlined in the Generic Hypotheses section is also desirable. However, for these to be effective, they will need to be versatile in order to account for variability in the nature of the receiving environment, the nature of the dredged material and disposal practises (see above).

Design and conduct of sampling programmes

Guidelines for the design of sampling programmes, and for sampling methods should largely be met by those devised for sewage-sludge disposal sites (Rees et al., 1990; see also Fig. 9). Special importance is attached to physico-chemical assessments of sediments in the receiving environment, and of transport pathways, for the design of effective biological sampling programmes. Pilot surveys employing remote sensing techniques such as side-scan sonar or towed photographic sledges are also be recommended.

The frequency of sampling will be determined on a site-specific basis, but should rarely exceed once per year.

Research and Development Requirements

The data-base on biological effects of UK dredgings disposal needs to be expanded, with regular sampling at major sites, so as to

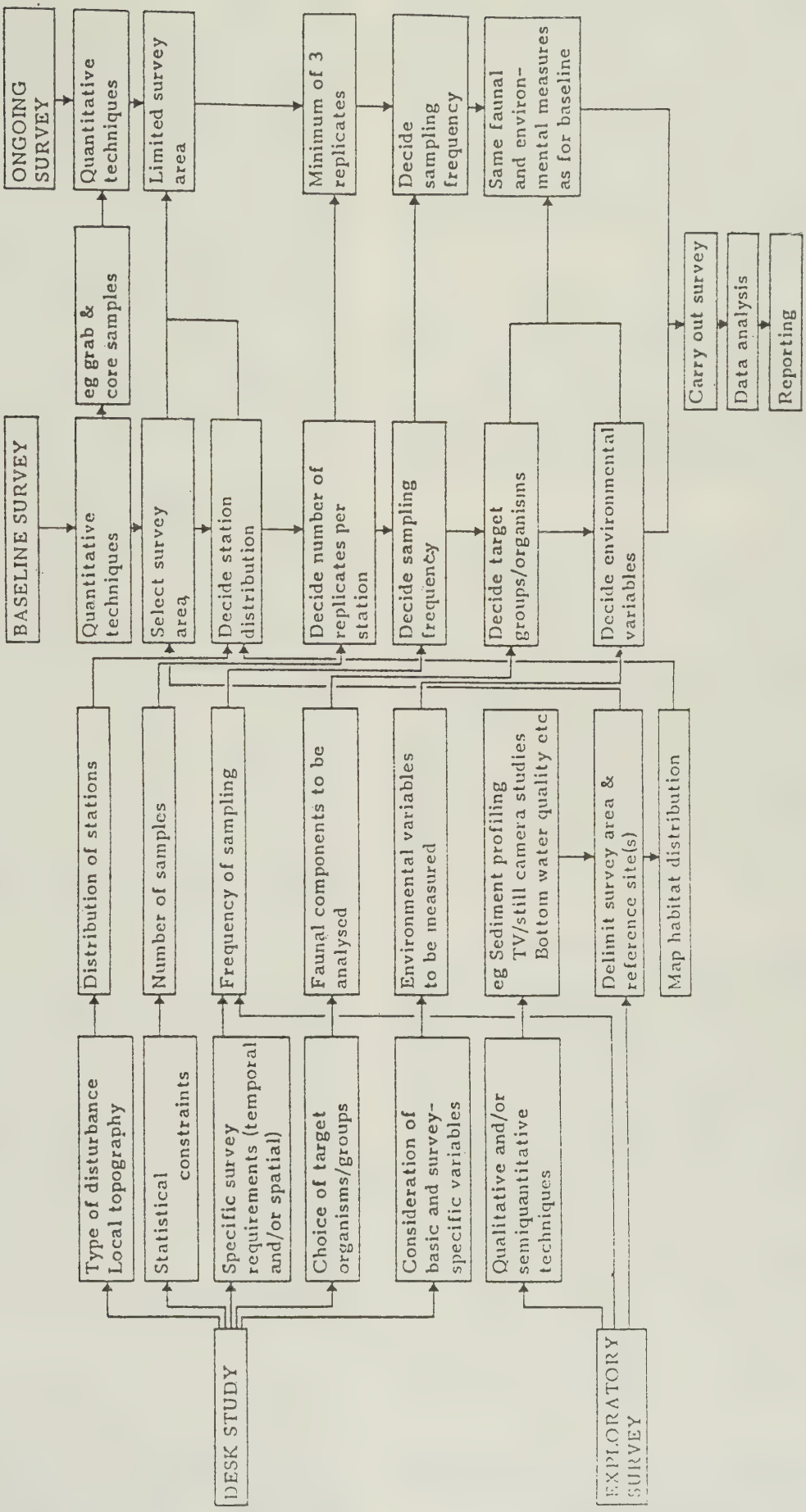


Figure 9. Model strategy summarising survey design. (Prepared by T H Pearson; reproduced from Rees et al., 1990).

facilitate the setting of realistic limits for acceptable change. The aim should be to encompass a range of material representative of the major arisings, and special attention should be paid to the identification of "indicator" species, investigations of biomass/production and (where appropriate) trophic links and the potential for contaminant bioaccumulation.

The use of experimental laboratory systems for the study of benthic changes in response to the dosing of dredged material from representative sources (or from those where contamination is perceived to be a particular problem) should also be considered as a predictive aid.

The value of remote imaging methods such as side-scan sonar and "Sediment Profile Imaging" (SPI) in the assessment of impacts, and in the design of benthos monitoring programmes, is recognised. Details of the latter are given in Appendix II; application to soft-sediment areas in receipt of dredgings should be supported, alongside validatory sampling of the benthos by established means. However, it should be remembered that this technique is not suitable for coarser deposits.

An up-dated review of the chemical quality and physical nature of dredged material at principal locations around the UK would strengthen the capability to predict the biological consequences at the point of disposal, and would also greatly assist in the development of practical guidelines for monitoring. Improved (and standardised) methods for characterising substrate types at the point of dredging would also be helpful in these respects, although the difficulties of obtaining representative samples of sufficiently small size for standard particle size analyses, from arisings which may amount to millions of tonnes, are recognised.

Further development of empirical models describing biological responses to different categories of dredged material is required.

Consideration should be given to the benefits (in biological terms) of the alternating use of disposal sites for certain major arisings, and of "capping".

Further attention should be given to the development of sampling protocols and quality assurance procedures for EQS compliance monitoring.

Discussion

The preceding account demonstrates that the nature and persistence of effects on the benthos arising from dredgings disposal may vary widely depending on the consistency of the material, its frequency of disposal and the nature of the receiving environment. For example, the disposal of uncontaminated sandy sediments which are similar in character to those at the disposal site may, in exposed inshore localities, simply mimic the consequences of periodic storm disturbance of sediments. On cessation of disposal, recovery may be relatively rapid (ie within months), through opportunistic recolonisation of sediments by short-lived species which are naturally common in the area. This recolonisation may proceed through planktonic recruitment (depending upon the time of year) or through re-distribution of adults. In contrast, the disposal of coarse material such as boulders or rock from capital dredging projects may induce long-term or even permanent changes to the habitat, depending on the capacity of water movements to erode or disperse the deposited material. In sufficient quantity, such material could present an obstacle to commercial trawling. However, it may also have the positive effect of locally increasing habitat variety, thereby promoting recolonisation by a wider range of species than those naturally present nearby, and of providing refuges for commercial fish or shellfish species (ie analogous to the creation of "artificial reefs").

Soft sediments comprise the bulk of material disposed of to sea, and arise largely from maintenance dredging of ports and harbours. Aside from the purely physical consequences, sediments which are enriched in organic matter, or carry an enhanced burden of contaminants (such as may arise from dredging within highly urbanised or industrialised estuaries) may require special

attention. Adverse effects of organically-enriched dredged material may arise where local accumulations induce anoxia within sediments through microbial action, resulting in the elimination of all but the most tolerant or adaptable benthic species. In areas where such dredged material is more widely dispersed after disposal, this may have a more positive effect of enhancing benthic populations without inducing major structural alterations to the sediments or fauna. It is conceivable that the presence of contaminants in dredged material could inhibit recolonisation through toxic effects, although the screening process prior to licensing for sea disposal should minimise the likelihood of such an occurrence. Of greater concern is likely to be the potential for contaminant bioaccumulation and then transfer through the food chain, especially if the disposal of dredgings results in a net enhancement of benthic organisms which are attractive as food for fish. Determination of the spatial scale over which such changes occur will be very important in deciding upon the potential significance of this route for contaminant transfer. In this respect, the wider issues of the management (and monitoring) of dredgings disposal as outlined in the guidelines of the Oslo Commission (1993) are highly relevant.

In practice, the assessment of effects will often be complicated in circumstances where the material to be disposed of varies over time in its nature or chemical "quality". This is a common occurrence at larger disposal sites around the United Kingdom. For example, a site may, at different times, receive dredgings from a wide range of localities which differ in their history of exposure to contamination and in the physical nature of the substratum. Nevertheless, in most cases it should be feasible to develop and then test an "impact hypothesis" through application of the (limited) available information on biological effects, along with guidelines for field sampling. As with other discharge types, sound sampling design and a thorough local knowledge of the waste-receiving area will be critical factors in determining the success of monitoring programmes, and hence of great importance to the effective management of dredged material disposal.

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Appendix I

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Appendix II

Sediment Profile Imaging

Sediment profile imaging (SPI), originally developed over 20 years ago as a rapid method of assessing faunal distributions with depth in the sediment, has recently, in conjunction with image analysis techniques, become more widely accepted as a rapid means of monitoring physico-chemical conditions and faunal distributions in the sediments (Rhoads and Germano, 1986 - these authors named the technique REMOTS, which is now a proprietary trade name in the USA). The basic technique involves obtaining a photograph, through a triangular prism inserted to about 150 mm depth in the sediment, of a vertical slice of sediment. Careful analysis of the features portrayed on the photograph can provide details of the sedimentary grain size, redox activity levels and faunal activity in the area. Deployment of such a system can be a means of rapidly surveying the sedimentary characteristics of a wide area and it has been used extensively in the USA for such purposes in recent years, particularly for monitoring dredged material disposal sites.

One of the major disadvantages in the use of SPI cameras is that no direct measurements can be made on the sediments from which images are obtained, and therefore the conclusions drawn from image analysis cannot be directly validated. This must be done by taking additional samples using core or grab samples which inevitably cannot be taken in exactly the same spot as the sediment profile image. An additional problem is the relative complexity and expense of an underwater SPI camera system. These difficulties have been addressed in a recent development by Grizzle and Penniman (1991), who describe the use of a coring device which obtains a sediment sample using a square transparent acrylic core tube through which a sediment profile image can be taken immediately on return of the sample to the surface. The core obtained can then be retained for the required physico-chemical and biological analyses to be carried out. This idea has now been extended further, by incorporating a square transparent core tube into a controlled entry, multiple coring instrument which is currently under development in Scotland. Such an instrument will provide sediment profile images in four vertical planes of a sample which can subsequently be analysed, together with two or three additional samples of sediment from adjacent areas at a measured distance from the original sample which can be used for physico-chemical and faunal analysis and for image validation. The instrument is intended to extend the range and improve the quality of environmental data obtained during benthic surveys, by providing a subsequent real-time validation of SPI images used to provide rapid initial survey information.

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